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The Empirics of Wetland Valuation: A Comprehensive Summary and a Meta-Analysis of the Literature†

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Abstract. Wetlands are highly productive ecosystems, providing a number of goods and services that are of value to people. The open-access nature and the public-good characteristics of wetlands often result in these regions being undervalued in decisions relating to their use and conservation. There is now a substantial literature on wetland valuation, including two meta-analyses that examine subsets of the available wetland valuation literature. We collected over 190 wetland valuation studies, providing 215 value observations, in order to present a more comprehensive meta-analysis of the valuation literature that includes tropical wetlands (e.g., mangroves), estimates from diverse valuation methodologies, and a broader range of wetland services (e.g., biodiversity value). We also aim for a more comprehensive geographical coverage. We find that socio-economic variables, such as income and population density, that are often omitted from such analyses are important in explaining wetland value. We also assess the prospects for using this analysis for out-of-sample value transfer, and find average transfer errors of 74%, with just under one-fifth of the transfers showing errors of 10% or less.

Key words: meta-analysis, valuation, value transfer, wetlands

JEL classifications: C53, D62, H23, Q20, Q25

1. Introduction

Wetlands are highly productive and valuable ecosystems. The public-good characteristics of many of the goods and services they provide often results in wetlands being undervalued in decisions relating to their use and conservation. Partly as a response to this situation, there is now substantial literature on wetland valuation (Barbier et al. 1997; Bardecki 1998; Kazmierczak

†This paper has not been submitted elsewhere in identical or similar form, nor will it be during the first three months after its submission to the Publisher.

2001). The empirical studies in this literature vary widely in their use of valuation techniques, the actual products and services being valued, and the type and geographical location of the wetlands being considered.

The resulting "flood of numbers" and the considerable cost associated with performing a study that assesses the value of a wetland has stimulated the use of research synthesis techniques, in particular meta-analysis (Stanley 2001; Smith and Pattanayak 2002; Bateman and Jones 2003). Meta-analysis is concerned with a quantitative analysis of statistical summary indicators reported in a series of similar empirical studies. In the case of wetland valuation, a standardized shadow price can be analyzed, such as the dollar value per year of 1 ha of wetland area. Meta-analysis extends beyond a state of the art literature review. Proponents of meta-analysis maintain that the valuable aspects of narrative reviews can be preserved in meta-analysis, and are in fact extended with quantitative features (Rosenthal and DiMatteo 2001). Some authors even refer to meta-analysis as a *quantitative* literature review (Stanley 2001).

Two wetland valuation meta-analyses already exist (Brouwer et al. 1999; Woodward and Wui 2001). These meta-analyses examine subsets of the available wetland valuation literature. They focus on temperate wetlands, and they consider a limited set of wetland services. Brouwer et al. (1999) restrict their sample to only contingent valuation studies. In addition, these studies do not include socio-economic and georeferenced information for the wetland sites in their respective meta-regression analysis. Consequently, there is scope for a more comprehensive meta-analysis of the valuation literature that includes tropical wetlands (e.g., mangroves), estimates from other valuation methodologies, other wetland services (e.g., biodiversity value), and estimates from more countries.

In this article, we provide a comprehensive overview of the empirical wetland valuation literature, reviewing virtually all studies that appeared over the last 25 years. We categorize the reported value estimates along several dimensions (such as wetland type, size, services, and valuation method), which leads to an exploratory synopsis of the determinants of wetland value. This analysis is complemented by a more rigorous assessment of the variation in wetland values by means of a meta-regression analysis. In this analysis we include socio-economic and georeferenced variables in the form of GDP per capita, population density, and latitude, as well as variables reflecting wetland and study characteristics. This potentially facilitates the use of "value transfer" to non-valued wetland sites as an alternative to primary valuation, although the validity and accuracy of such a value transfer has been questioned (Downing and Ozuna 1996; Brouwer and Spaninks 1999; Brouwer 2000). Following up on, among others, Rosenberger and Loomis (2000), and Bateman and Jones (2003) we explicitly investigate the validity, the efficiency, and the robustness of value transfers based on the meta-analysis of wetland values.

The organization of this article is as follows. Section 2 outlines the definition and typology of wetlands used in this article, the wetland functions that are utilized by humans, and the valuation methods that are applied to value various wetland services. This section also discusses the heterogeneity of the value estimates. Section 3 gives an overview of the empirical wetland valuation literature, and presents the results of an exploratory analysis. We show the resulting descriptive statistics and cross-tabulations against, for instance, type of wetlands, wetland services, and valuation methodology. Section 4 describes the setup for a meta-regression, specifically the specification and functional form of the meta-regression function. This section also gives the regression output and an interpretation of the results. In Section 5, we explore the validity, efficiency and robustness of using a meta-valuation function that includes socio-economic and georeferenced information in a value transfer exercise. Finally, Section 6 concludes and provides suggestions for future research and policy.

2. Wetland Types, Functions and Values

A widely agreed upon, precise definition of what constitutes a wetland is not available. However, in "The Convention on Wetlands," a UNESCO-based intergovernmental treaty on wetlands adopted in the Iranian city of Ramsar, in 1971 (more commonly known as the "RAMSAR Convention") provides a broad characterization. The RAMSAR convention on wetlands defines wetlands very broadly as (Article 1.1):

> areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres,

and points out (in Article 2.1) that wetlands:

> may incorporate riparian and coastal zones adjacent to the wetlands, and islands or bodies of marine water deeper than six metres at low tide lying within the wetlands.

Depending on interpretation, this very inclusive definition encompasses a large number of ecosystem types. As of 2004, the "RAMSAR Convention" includes 1369 wetland sites, located in 139 countries throughout the world, although the location of the sites is strongly skewed towards Western Europe (see http://ramsar.org/sitelist.pdf). The RAMSAR-sites cover over 120.5 million hectares of wetland. In this study, we use the same definition and we specifically classify wetlands into five types: mangroves, unvegetated sediment, salt/brackish marsh, freshwater marsh, and freshwater woodland.

Depending partly on wetland type, wetlands provide a number of goods and services that are of value to humans (Barbier 1991). The services

Table I. Ecological wetland functions, economic goods and services, types of value, and applicable valuation methods

Ecological function	Economic goods and services	Value type	Commonly used valuation method(s)[a]
Flood and flow control	Flood protection	Indirect use	Replacement cost Market prices Opportunity cost
Storm buffering	Storm protection	Indirect use	Replacement cost Production function
Sediment retention	Storm protection	Indirect use	Replacement cost Production function
Groundwater recharge/discharge	Water supply	Indirect use	Production function, NFI Replacement cost
Water quality maintenance/nutrient retention	Improved water quality	Indirect use	CVM
	Waste disposal	Direct use	Replacement cost
Habitat and nursery for plant and animal species	Commercial fishing and hunting	Direct use	Market prices, NFI
	Recreational fishing and hunting	Direct use	TCM, CVM
	Harvesting of natural materials	Direct use	Market prices
	Energy resources	Direct use	Market prices
Biological diversity	Appreciation of species existence	Non-use	CVM
Micro-climate stabilization	Climate stabilization	Indirect use	Production function
Carbon sequestration	Reduced global warming	Indirect use	Replacement cost
Natural environment	Amenity	Direct use	HP, CVM
	Recreational activities	Direct use	CVM, TCM
	Appreciation of uniqueness to culture/ heritage	Non-use	CVM

Source: with modifications adapted from Barbier (1991, 1997), Brouwer et al. (1999), and Woodward and Wui (2001).
[a]Acronyms refer to the contingent valuation method (CVM), hedonic pricing (HP), net factor income (NFI), and the travel cost method (TCM).

provided by wetlands are derived from, but should not be confused with, their ecological and physical functions. Table I lists the main ecological/ physical functions of wetlands, and their associated economic goods and services.

The range of services provided by wetlands is partly related to direct geophysical processes, such as sediment retention and the provision of flood and storm buffering capacity, but it extends to wider climatologic, biological, and socio-cultural functions, including impacts on local and global climate change and stabilization, preservation of biodiversity, and the provision of natural environmental amenities. In addition, wetlands provide ecological processes enabling the extraction of goods and services in the form of natural resources such as water, fish and other edible animals, wood, and energy, and they provide the natural surroundings for recreational activities (see Larson et al. 1989; Barbier 1991, 1997; Brouwer et al. 1999; Woodward and Wui 2001).

The economic values associated with these wetland goods and services can be categorized into distinct components of the total economic value according to the type of use. Direct use values are derived from the uses made of a wetland's resources and services, for example wood for energy or building, water for irrigation and the natural environment for recreation. Indirect use values are associated with the indirect services provided by a wetland's natural functions, such as storm protection or nutrient retention. Non-use values of wetlands are unrelated to any direct, indirect or future use, but rather reflect the economic value that can be attached to the mere existence of a wetland (Pearce and Turner 1990).

These components of the total economic value of wetlands often do not accrue to the owner of the wetland, and as a result, important wetland values are often overlooked in decision-making on wetland conversion (see Cummings and Harrison 1995). Some wetlands goods and services may be traded directly in well functioning markets and therefore have readily observable (marginal) values. However, due to market failures resulting from undefined property rights or the (quasi) public good characteristics of some wetland services, many valuable wetland services may not be traded directly or even indirectly through markets. Examples of wetland services that are indirectly traded through markets may include the amenities associated with housing located near to wetlands or water supply provided to agriculture. Wetland services that may not even be indirectly traded through markets include bequest and existence values. In cases where the values of important wetland services are not observable in well functioning markets, a number of non-market valuation methods may be applied to estimate economic values.

A diverse range of valuation methods have been applied to value wetland services, including the contingent valuation method (e.g., Farber 1988; Bateman and Langford 1997), hedonic pricing (e.g., Lupi et al. 1991; Doss and Taff 1996), travel cost method (e.g., Ramdial 1975; Cooper and Loomis 1993), production function approach (e.g., Acharya and Barbier 2000; Bell 1997), net factor income approach (e.g., Amacher et al. 1989; Schuijt 2004), total revenue estimation (e.g., Costanza et al. 1989; Raphael and Jaworski 1979), opportunity cost (e.g., Leitch and Hovde 1996; Sathirathai and Barbier 2001), and replacement cost (e.g., Breaux et al. 1995; Emerton and Kekulandala 2002). The applicability of each of these methods depends largely on the wetland service being valued and the type of value associated with it (Freeman 2003). Table I lists the valuation methods next to the wetland services and value types that they are commonly used to value. It must be noted that these valuation methods differ considerably in terms of the welfare measures that they estimate (see Freeman 2003; Kopp and Smith 1993; Carson et al. 1996). This source of heterogeneity in the meta-data may lead to problems of non-comparability between estimated values and we need to be wary of comparing inconsistent concepts of economic value (Brouwer 2000; Smith and Pattanayak 2002).

Table II lists the valuation methods used within the wetland valuation literature together with a short description of each method and the welfare measure that it estimates. The contingent valuation method is the only method capable of estimating non-use values and by directly asking respondents to state their WTP or WTA for (hypothetical) changes in environmental quality

Table II. Valuation methods and associated welfare measures

Valuation Method	Short description	Welfare measure
Contingent valuation	Hypothetical questions to obtain WTP	Compensating or equivalent surplus
Travel cost	Estimate demand (WTP) using travel costs to visit site	Consumer surplus
Hedonic pricing	Estimate WTP using price differentials and characteristics of related products	Consumer surplus
Production function	Estimate value as an input in production	Producer and consumer surplus
Net factor income	Assign value as revenue of an associated product(s) net of costs of other inputs	Producer surplus
Replacement cost	Cost of replacing the function with an alternative technology	Value larger thanthe current cost of supply
Opportunity cost	Value of next best alternative use of resources (e.g., agricultural use of water and land)	Consumer surplus, producer surplus, or total revenue for next best alternative
Market prices	Assigns value equal to the total market revenue of goods/services	Total revenue

or quantity it provides estimates of the technically precise welfare measures of compensating and equivalent surplus. The hedonic pricing and travel cost methods estimate the Marshallian consumer surplus, which approximates, and is bounded by, the compensating variation (CV) and equivalent variation (EV) welfare measures. For relatively small price changes, the error of approximation between consumer surplus, CV and EV is small (Willig 1976). For large price changes, however, (e.g., when considering a price change large enough to drive the quantity demanded to zero) the error can be substantial (Freeman 2003). In response to this potential error there are now numerous travel cost and hedonic pricing studies that attempt to estimate Hicksian measures of consumer surplus (see for example Shaw and Ozog 1999). There is empirical evidence to suggest that revealed preference and contingent valuation methods produce broadly similar value estimates (Bateman et al. 2004).[1] The production function approach estimates changes in consumer and producer surplus resulting from quantity or quality changes in an environmental good that is used as an input in a production process. If the price of output is unaffected by the environmental change (i.e., if demand for the good is perfectly elastic), only producer surplus is affected. The net factor income approach also estimates changes in producer surplus by subtracting the costs of other inputs in production from total revenue, and ascribes the remaining surplus as the value of the environmental input.

The remaining valuation methods do not have a sound basis in welfare theory and therefore may be expected to over- or underestimate economic values. The bold line across the middle of Table II indicates the distinction between methods that have sound underpinnings in welfare economics and those that do not. The total revenue approach simply estimates values as the total revenue received from the sale of goods or services derived from the environmental resource in question. This approach ignores the cost of all other inputs in the production of these goods and services and will therefore tend to overestimate producer surplus. The opportunity cost approach takes the value of the next best alternative use of the resources used to provide the ecosystem function being valued. This reflects the cost of supplying the good or service and not the surplus associated with its use. The replacement cost approach places values on ecosystem services by estimating the cost of replacing them. This approach is based on the assumption that if individuals incur costs to replace ecosystem functions, then the lost services must be worth at least what people are willing to pay to replace them. Replacement costs are not based on social preferences for ecosystem services, or individuals' behavior in the absence of those services, and are unlikely to approximate consumer and producer surpluses. Even with evidence to suggest that society is willing to pay for an identified least cost replacement for lost ecosystem services this is not a theoretically valid estimator of ecosystem service value.

The diversity in welfare measures being estimated makes it necessary to clearly distinguish between the different valuation techniques in the meta-analysis. Although we may have *a priori* expectations as to the direction of any bias associated with each valuation method,[2] it is not possible to make sensible adjustments to the observed valuation estimates to correct for these biases. The differences in values estimated through each method are examined initially in Section 3 and using a meta-regression in Section 4.

3. Overview of the Empirical Wetland Valuation Literature

Responding to the fact that the value of wetland services are often not known and therefore not included in decisions regarding wetland use and conservation, there is now a large number of studies attempting to value the partial or total economic value of numerous wetland sites. For the purposes of conducting a meta-analysis of wetland values, we have attempted to collect as much of the available literature as possible. The methods employed in literature retrieval included searching electronic journal databases, libraries, existing bibliographies and databases of wetland valuation studies, and contact with authors and relevant agencies.[3] In total, 191 studies related to wetland valuation were collected. The earliest of these studies is a 1969 CVM estimation of consumer surplus for wildfowl hunting in wetlands of the US Pacific western flyway (Hammack and Brown 1974). We found eight wetland valuation studies that were conducted in the 1970s, seven of which are for US wetlands. Twenty-five studies were found from the 1980s, 124 from the 1990s and 32 from 2000 or later. As well as an apparent upward trend in the number of valuation studies being conducted, there has also been a diversification in the valuation methodologies used and the geographic location of the wetlands being valued. It should be noted that the apparent trend of increasing wetland valuation work might in part be due to the enhanced availability of more recent valuation studies, i.e. through internet publication and electronic journals.

The studies that have been collected cover various publication outlets, including journal articles, project reports, dissertations, and book chapters. This literature is very diverse in terms of the objectives of the research being presented. Generally, the literature can be categorized into three groups according to the primary focus of the study. First, some studies merely estimate one or more values for a specific wetland site (e.g., Costanza et al. 1989; Lant and Roberts 1990; Cooper and Loomis 1991; Barbier and Strand 1998, Emerton et al. 1998; Klein and Bateman 1998; Acharya 2000). Second, some studies review or compare already existing wetland valuations (e.g., Anderson and Rockel 1991; Gren and Soderqvist 1994; Barbier et al. 1997; Dixon and Lal 1997; Bardeki 1998). Third, some studies develop a specific methodological innovation for non-market valuation of wetlands (e.g., Barbier 1991; Creel and Loomis 1992; Dalecki et al. 1993; Gren et al. 1994; Bateman and Langford 1997; Ellis and Fisher 1987; Haab and McConnell

1997; Pate and Loomis 1997). Obviously, many studies combine elements of these three categories to some degree.

In addition, there are some important distinctions to be made between studies within these three general categories. Within the group of studies that primarily attempt to estimate values for specific wetland sites, there are some that estimate values for alternative wetland management strategies (e.g., Freeman 1991; Ruitenbeek 1992; Bann 1997; Barbier and Thompson 1998; Janssen and Padilla 1999; Van den Bergh et al. 2001). Other studies use value transfer rather than primary valuation techniques to value a specific wetland (e.g., Farber and Costanza 1987; Dahuri 1991; Farber 1992; Gren 1995; Dharmaratne and Strand 2002). Within the group of studies that focuses on methodological issues in wetland valuation, some empirically test different survey or estimation techniques using real or hypothetical data whereas others are purely conceptual (e.g., Bergstrom and Stoll 1993; Bystrom et al. 2000; Turner et al. 2000).

Of the 191 collected studies, 80 contained suitable and sufficient information for the purposes of comparison in a statistical meta-analysis.[4] From these 80 studies, we were able to extract 215 separate observations of wetland value. The maximum number of observations taken from one study is 10 and the average number is approximately 2.7. Care was taken not to double count value estimates that are reported in more than one study, or to include estimates that were derived through value transfer from studies also included in our data set. The main reasons for not including information from a study are that it either reports already published results or is focused on methodological issues rather than primary valuation.

The observations in our data set are from 25 countries and all continents are represented. Figure 1 presents a map of the spatial distribution of the wetlands in our database, for which a value estimate is available.

It should be noted that the geographical distribution of observations in our sample reflects the practice and availability of natural resource valuation studies rather than the distribution of wetlands. North America, for example, is particularly well represented with half our data set comprising of observations from the US and Canada. The number of wetlands represented in our data set, however, is less than the number of observations because multiple observations are taken from each study, which typically only consider one wetland, and several studies have valued the same wetland. Although we have 16 separate valuation observations for Africa, it can be seen from Figure 1 that these are for only five wetlands. For Australasia, on the other hand, we have seven observations for five different wetlands.

Figure 2 shows the number of wetland value observations in our sample categorized according to wetland type, wetland service, and valuation method. The valuation literature has clearly focused on freshwater wetlands.

Figure 1. Geographic location of wetlands for which value estimates are available.

In addition to the large number of observations for North American freshwater wetlands, there is also a marked focus on mangrove valuation in Asia. This valuation effort has been prompted by the large-scale conversion of mangroves to other land uses, such as shrimp ponds, and logging in recent decades.

A large number of different wetland services have been valued in the literature, although not all of the services identified in Table I have been valued (e.g., carbon sequestration and micro-climate stabilization). In order to reflect the distinctions that are generally made between wetland services in the valuation literature we have categorized wetland services in our database slightly differently from the list in Table I. This involved combining some services (e.g., flood control and storm buffering) and separating others (e.g., recreational hunting and fishing). The wetland service categories that we used are: flood control and storm buffering, water supply, water quality, habitat and nursery service (specifically support for commercial fisheries and hunting), recreational hunting, recreational fishing, amenity and other recreational uses, materials, fuel wood, and biodiversity. The number of observations for each wetland service is presented in Figure 2. Most studies value only one particular wetland service rather than all services provided by the wetland in question. There is, however, also a significant number of studies that value two or more services of a wetland and a small number that estimate total economic value, i.e., referring to all important goods and services (for example, Blomquist and Whitehead 1998; Leitch and Hovde 1996).

A broad range of valuation methodologies has been applied to value wetlands. The numbers of observations for each method are shown in Figure 2. The method most commonly used in the literature has been to observe the

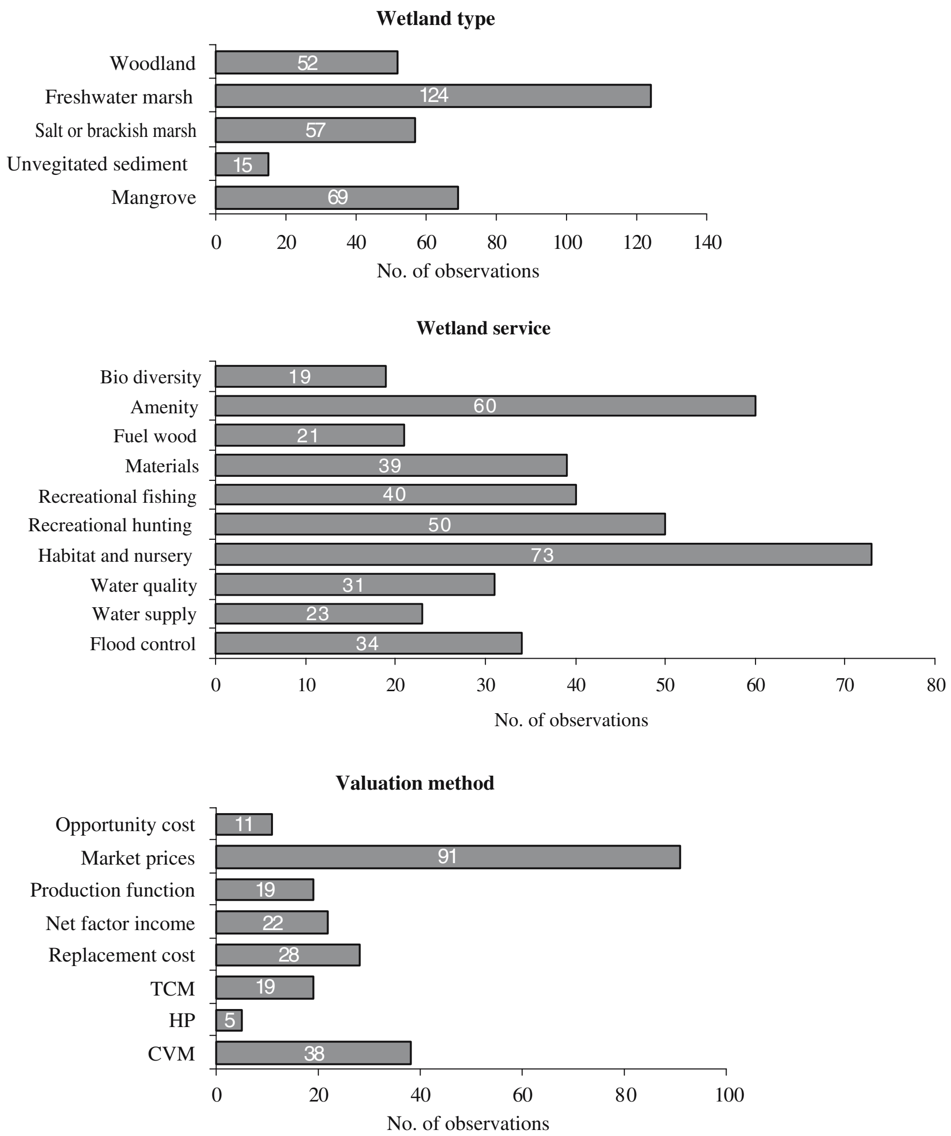

Figure 2. Number of observations for each wetland type, wetland service, and valuation method.

market prices of products related to wetland services and then ascribe the total revenue from the sale of such products as the value of the wetland. Contingent valuation has also been widely used. As expected, the different valuation methodologies have been applied to value different wetland services. CVM, hedonic pricing and TCM have been applied to value amenity and recreational values, replacement cost has largely been used to value the role of wetlands in improving water quality, and the production function approach has been used to value the habitat and nursery services of wetlands. The market price approach has been used to value most wetland services.

Wetland values have been reported in the literature in many different metrics, currencies and referring to different years (e.g., WTP per household per year, capitalized values, marginal value per acre, etc). In order to enable comparison between these values we standardized them to 1995 US dollars per hectare per year, following Woodward and Wui (2001).[5] In standardizing wetland values we faced the problem of distinguishing between average and marginal values, both of which can be expressed as a monetary value per hectare. The majority of wetland valuation studies have estimated total or average wetland values but there is also a large number of estimates of marginal wetland values. Small changes in wetlands should be valued using marginal changes (Batie and Shabman 1979) whereas average values may be useful for comparing the aggregate value of a wetland area relative to the size of the area (Bergstrom et al. 1990). Expressing wetland values in per hectare terms gives the impression that each hectare in a wetland is equally productive, i.e., that wetlands exhibit constant returns to scale or equivalently that the marginal wetland value is equal to average wetland value. Without being able to convert marginal values to average values or vice versa, we assume exactly this. This assumption is examined later on in the discussion on whether wetlands exhibit increasing or decreasing returns to scale.

Standardizing wetland values to WTP per person as in Brouwer et al. (1999) was not possible because several of the valuation methods used in the literature (e.g., NFI, opportunity cost, replacement cost and market prices) do not produce WTP estimates. WTP per person or household on the other hand could be converted to US\$ ha^{-1} yr^{-1} given information on the wetland area and the relevant population size. Using an annual dollar value per unit of area may also facilitate the use of meta-analysis results for value transfer because in most cases it is more straightforward to transfer values to a given wetland area than to the relevant number of people that are willing to pay for wetland conservation.

For our data set the average annual wetland value is just over 2800 US\$ per hectare. The median value, however, is 150 US\$ ha^{-1} yr^{-1}, showing that the distribution of values is skewed with a long tail of high values. As expected, the mean and median values of wetlands vary considerably by continent, wetland type, wetland service and valuation method used. Figure 3 presents the mean and median wetland value for each continent, wetland type, wetland service, and valuation method respectively. The information contained in this Figure does not account for the variation in wetland values that is explained by variation in other important variables. In order to examine the importance of each variable in explaining the variation in wetland values while accounting for variation in other variables we use a meta-regression as set out in the following section. This graphical representation of the data, however, helps

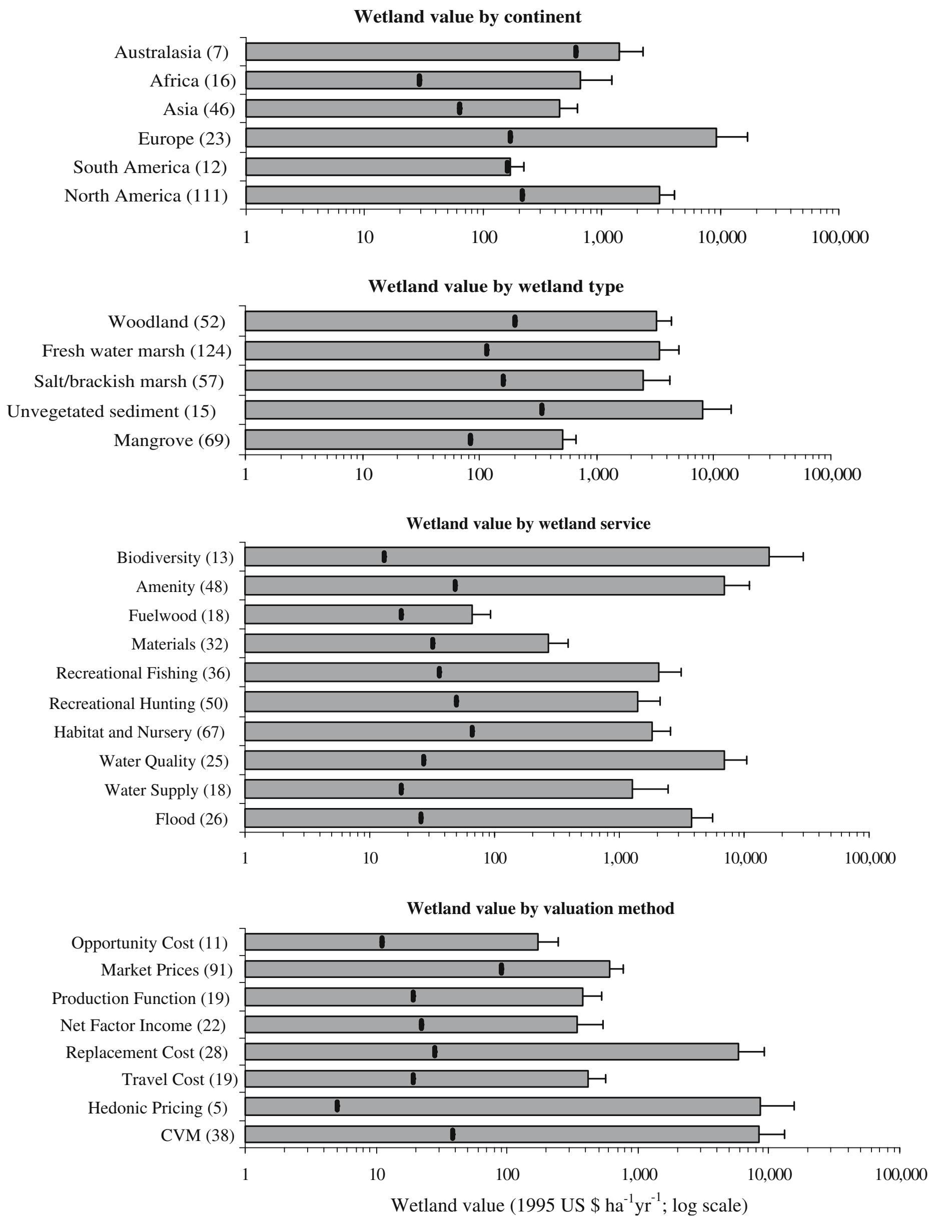

Figure 3. Mean and median wetland values for each continent, wetland type, wetland service, and valuation method. The number of observations for each category is in parentheses. The bars represent the means, the error bars represent the standard error of the mean, and the black dots represent the medians.

to give an initial understanding of the determinants of variation in wetland values found in the literature.

Figure 3 shows that average wetland values are highest in Europe, followed by North America, Australasia, Africa, Asia, and finally South America. It also shows that the wetland type unvegetated sediment has the highest average value of just over 9000 US$ ha^{-1} yr^{-1}. Mangroves have the lowest average value of just over 400 US$ ha^{-1} yr^{-1}. In terms of median values the variation is much less, suggesting that different wetland types do not have largely different values. In our sample, the biodiversity service of wetlands has the highest average value (17,000 US$ ha^{-1} yr^{-1}), and the use of wetlands for collecting fuel wood and other raw materials has the lowest values (73 and 300 US$ ha^{-1} yr^{-1}, respectively). Studies that have used the contingent valuation method (CVM) have produced the highest estimates of wetland value, followed by the replacement cost method and hedonic pricing. The lowest value estimates are produced by the opportunity cost and production function methods. These differences in values produced by alternative valuation methodologies may in part be explained by the application of these methods to value different wetland services (as described above), and also be due to the differences and biases in welfare measures that each method estimates.

Another wetland characteristic that we may expect to determine wetland value is its area. There is no clear *a priori* expectation of the sign of this relationship given on the one hand that there may be diminishing marginal returns to most wetland services as wetland size increases, but on the other hand some ecological functions require minimum thresholds of habitat area which suggests that wetland values may increase with size. Figure 4 plots wetland value by wetland area and reveals a possible negative relationship between the two. The trend line represents an estimated least squares regression equation. The coefficient on wetland area is not significant. The wetlands for which value estimates are available are generally medium to large size wetland areas. This does have implications for the extent to which economies of scale can be estimated reliably; see also Woodward and Wui (2001) who conclude that over a large range of sizeable wetlands constant returns to scale are apparent.

In addition to the wetland characteristics and valuation methods that are examined above, we would expect that the value of a wetland is determined by the socio-economic characteristics of its location. Information regarding per capita income of the relevant population using each wetland was generally not available in the valuation studies so we inputted this information from other sources.[6] Figure 4 also plots wetland value per hectare per year by GDP per capita. As expected there appears to be a positive relationship between the two. The coefficient on GDP per capita in the simple regression is significant at the 10% level. We also included population density information for each wetland site in the database in order to examine the influence of population

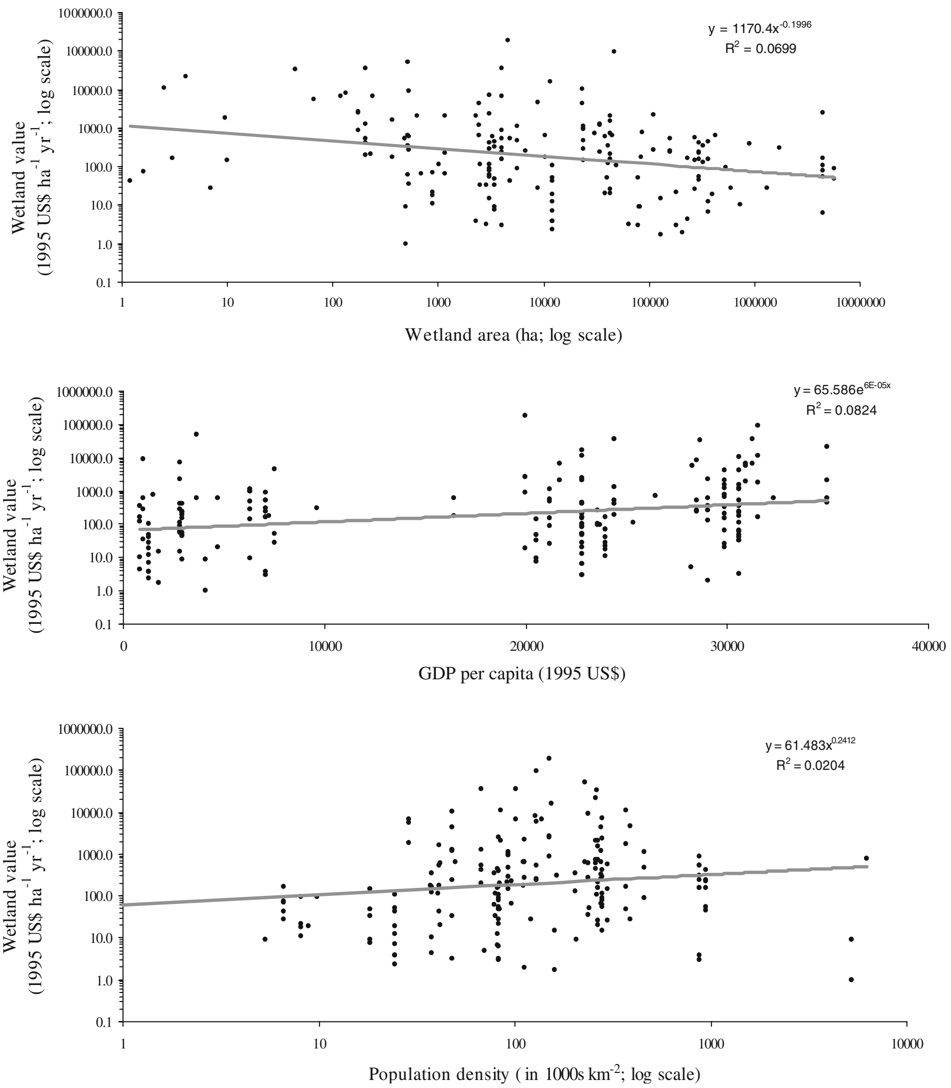

Figure 4. Wetland value per hectare per year plotted against wetland area, GDP per capita, and population density.

density on wetland values.[7] Our expectation is that wetlands have a higher value in areas with higher population density as most wetland services are related to direct or indirect human use. The spatial relationship between wetlands and centers of population is of course important in determining the use made of wetland services. This spatial relationship will vary with a number of factors including wetland service, transportation availability, physical barriers, and cultural norms. For example, in the case of the recreational use of a wetland we would expect that in the US the distance between a population center and the wetland would be of less importance than in a developing country due to transportation availability and habits. Consequentially, the "catchment" area of population that might use a wetland would be much larger in the US. We were not able to capture all of these considerations in the data but use population densities for 50-km radius zones around each wetland site. Figure 4 plots wetland value by population density. There is an apparent positive relationship between the two, although the estimated coefficient on population density in the trend line equation is not significant.

4. Meta-Regression

The above exploratory analysis of the available data in the wetland valuation literature does of course not allow for interactions between the various explanatory variables. In order to attain marginal effects – given the interference of potentially relevant intervening characteristics – we use meta-regression analysis to assess the relative importance of all potentially relevant factors simultaneously. The dependent variable in our regression equation is a vector of values in US$ per hectare per year in 1995 prices, labelled y. The explanatory variables are grouped in three different matrices that include the study characteristics in X_s (i.e., valuation method, marginal value), the wetland physical and geographical characteristics in X_p (i.e., wetland type, services, area, urban, continent, latitude, and RAMSAR proportion), and the socio-economic characteristics in X_e (i.e., GDP per capita, and population density). The model fit was considerably improved, and heteroskedasticity was mitigated, by using the logarithms of the dependent variable, GDP per capita, population density, and wetland size. The estimated model is, in matrix notation:

$$\ln(y) = a + X_s b_s + X_p b_p + X_e b_e + u$$

where a is the usual constant term, u a vector of residuals (assuming well behaved underlying errors), and the vectors b contain the estimated coefficients on the respective explanatory variables.[8] The regression results are presented in Table III, using White-adjusted standard errors because the Breusch–Pagan test still indicates that the model is heteroskedastic. Multicollinearity was tested for and judged not to be a serious problem.[9] The adjusted R^2 value of 0.45 is reasonably high, and indicates that close to half the variation in wetland value is explained by variation in our explanatory variables. In this (largely) semi-log model, the coefficients measure the constant proportional or relative change in the dependent variable for a given absolute change in the value of the explanatory variable. For the explanatory variables expressed as logarithms, the coefficients should be interpreted as elasticities, that is, the percentage change in the dependent variable given a (small) percentage change in the explanatory variable.

Table III. Meta-regression results[a]

Category	Variable[b]	Coefficient	Standard error
	Constant	−6.98	4.67
Socio-economic	GDP per capita (log)	1.16**	0.46
	Population density (log)	0.47***	0.12
Geographic characteristics	Wetland size (log)	−0.11**	0.05
	Latitude (absolute value)	0.03	0.07
	Latitude squared	−0.0007	0.0010
	South America	0.23	1.19
	Europe	0.84	0.92
	Asia	2.01	1.34
	Africa	3.51**	1.52
	Australasia	1.75*	0.94
	Urban	1.11**	0.48
Valuation methods	CVM	1.49**	0.73
	Hedonic pricing	−0.71	1.54
	TCM	0.01	0.65
	Replacement cost	0.63	0.81
	Net factor income	0.19	0.61
	Production function	−1.00	0.75
	Market prices	−0.04	0.53
	Opportunity cost	−0.03	0.72
Type value	Marginal	0.95*	0.48
Wetland type	Mangrove	−0.56	0.82
	Unvegetated sediment	0.22	1.09
	Salt/brackish marsh	−0.31	0.42
	Fresh marsh	−1.46**	0.59
	Woodland	0.86**	0.42
Wetland service	Flood control	0.14	0.55
	Water supply	−0.95	0.71
	Water quality	0.63	0.74
	Habitat and nursery	−0.03	0.35
	Hunting	−1.10**	0.43
	Fishing	0.06	0.36
	Material	−0.83**	0.42
	Fuelwood	−1.24***	0.45
	Amenity	0.06	0.39
	Biodiversity	0.06	0.81
RAMSAR	RAMSAR proportion	−1.32*	0.70
	n	202	

Table III. Continued

Category	Variable[b]	Coefficient	Standard error
R^2-adjusted		0.45	
F		5.50***	
Breusch–Pagan		51.46***	

[a] OLS results with White-adjusted standard errors. The Breusch–Pagan test concerns heteroskedasticity and is χ^2 distributed with 36 degrees of freedom. Significance is indicated with ***, **, and * for the 1, 5, and 10% level, respectively.
[b] The valuation methods, wetland types and wetland services are not strictly non-overlapping variables. In other words, some wetlands provide more than one service, and comprise smaller areas of different types. There is also not a one-to-one correspondence between an observed value and the use of a specific valuation method. Consequently, there is no need for the omission of one of the categories in order to avoid perfect collinearity.

Regarding the influence of wetland type on the wetland value, differences in value associated with different wetland types are indicated by the coefficients on these dummy variables. Two of these coefficients are significantly different from zero suggesting that fresh marshes have the lowest value as compared to the average and woodland wetlands have the highest value. This result was not apparent from Figure 3, but the latter result was merely based on bivariate comparisons.

On the issue of whether wetlands exhibit increasing or decreasing returns to scale, the coefficient on the wetland size variable is small and negative, as well as significant. This suggests that there are significant decreasing returns to scale. We correct for the fact that marginal values may be significantly different from average values, which is shown to be the case. Specifically, marginal values are almost twice as high as compared to average values. The decreasing return result confirms the findings of Woodward and Wui (2001), who observe decreasing returns to scale for wetlands at the level of −0.17 and −0.29 for a comparable function. It should be noted that the double-log specification induces the returns to scale to decline geometrically with size (see Woodward and Wui 2001, pp. 267–268), so that the elasticity approaches zero with increasing size.

The differences in wetland values resulting from the availability of different services were touched on above. Wetland services that involve the provision of direct use natural resources, such as fuel wood and other materials tend to have lower than average values. Somewhat surprisingly, wetlands that provide recreational hunting opportunities also tend to have lower than average values.

Another unexpected result given the initial analysis of Figure 3 is that North American wetlands tend to have lower values than wetlands located in other continents. North America is included in the constant term of our model and the coefficients on the dummy variables indicating the other continents are all positive but only significant for Africa and Australasia. We can only speculate on the reason for this. One possible explanation for wetlands receiving a lower value in North America is the relative abundance of substitute natural areas, particularly in comparison with Europe. One should note that these results are obtained correcting for differences in latitude. We hypothesized that the value of wetlands might be related non-linearly (following a parabolic shape) to the absolute distance from the equator. This, however, is not apparent in the estimation results.

For the socio-economic variables that we were able to include in our model, the results confirm our expectations. The coefficient on the GDP per capita variable is positive and highly significant – suggesting a slightly elastic effect of income on the value of wetland services. The interpretation of the result is that a 10% increase in GDP per capita results in roughly a 12% increase in wetland value. There is also a positive and significant relationship between population density and wetland value as described above. This relationship, however, is inelastic, but this may very well be due to the dummy variable "Urban" that is also included. On average, urban wetlands have a value that is significantly higher than rural wetlands.

The results for the valuation methodology dummy variables show that value estimates from contingent valuation, replacement cost, the travel cost method and NFI methods are higher than estimates from other valuation methods. The only statistically significant result, however, pertains to CVM, which show the highest values as compared to the other valuation methods. This result is in contrast with the findings of Woodward and Wui (2001), who observed that the hedonic pricing and the replacement cost method produce higher values than CVM.

Finally, the variable referring to a comparison between RAMSAR and other sites shows an interesting outcome. We use a variable operationalized as the proportion of the wetland that is designated a RAMSAR site, and it shows up indicating significantly lower values for RAMSAR sites. Possible explanations for this result might be that certain uses of these wetlands are restricted and therefore not valued, or that WTP bids for the conservation of these sites are affected by respondents' knowledge that they are already protected.

5. Value Transfer

There remains the question of whether the results from this meta-analysis can be used for value transfer, that is, the prediction or estimation of the value of a wetland given knowledge of its physical and socio-economic characteristics.

There is substantial academic and policy interest in the potential for and validity of value transfer as it offers a means of estimating monetary values for environmental resources without performing relatively time consuming and expensive primary valuation studies (see Florax et al. 2002).

There are two general approaches to value transfer: direct value transfer and function value transfer. The first involves simply transferring the value(s) estimated in one or more primary studies to the policy site in question. Ideally, the study site and policy site should be similar in their characteristics or adjustments should be made to the transferred value to reflect differences in site characteristics (Brouwer 2000). The second approach involves transferring values to a policy site based on its known characteristics using a value transfer function, possibly estimated through a meta-regression. Rosenberger and Phipps (2001) identify the important assumptions underlying the use of a value function for value transfer:

(1) there exists a valuation function that links the values of a resource with the characteristics of the relevant markets and sites across space and time, and from which values for specific characteristics can be inferred,
(2) differences between sites can be captured through a price vector,
(3) values are stable over time, or vary in a systematic way, and
(4) the sampled primary valuation studies provide "correct" estimates of resource value.

It is generally accepted that function transfers perform better than direct value transfers for a number of reasons. Firstly, information from a larger number of studies is used. Secondly, methodological differences between primary valuation studies can be controlled for. And thirdly, explanatory variables can be adjusted to represent the policy site (Bateman and Jones 2003). Rosenberger and Phipps (2001) review a number of studies that test the relative performance of direct value transfer and function value transfer (see for example Loomis 1992; Parsons and Kealy 1994; Brouwer and Spaninks 1999). The general conclusion is that meta-analysis value transfer functions perform better than other approaches (see also Engel, 2002).

For a number of reasons value transfer may result in substantial 'transfer errors', particularly when the characteristics of the site to which values are being transferred are not well represented in the data underlying the estimated value function (Brouwer 2000). Another reason for error might be that the characterization of wetland types and wetland services is oftentimes rather crude. The use of dummy variables to characterize types and services does not capture the true variation in these characteristics. Similarly, it is difficult to capture important quality and quantity differences in provision of services across sites. Unfortunately, we cannot overcome this in our value transfer experiment, but we have instead focused on including non-sample information such as GDP per capita and population density. To the extent that such information is relevant, it may increase the accuracy of value transfer estimates. We have also attempted to be as comprehensive as possible in our collection of valuation studies but clearly due to the limited availability of studies, our sample focuses on certain wetland types, services and locations. This raises the question of the validity of value transfer to policy sites in countries that are not represented in the data (Shrestha and Loomis 2001). Another important source of transfer error is related to errors in the primary valuation studies from which the value transfer function is derived (Woodward and Wui 2001). As described in Section 2, there are a number of biases associated with each valuation method that may result in mis-estimation of "true" resource values and thereby introduce a source of error in estimating a value function.

As a first step, before actually performing a value transfer, we looked at the in-sample forecast performance of our model. As an indicator we used the Mean Absolute Percentage Error (MAPE), which is defined as $|(y_{\text{obs}}-y_{\text{est}})/y_{\text{obs}}|$. For our sample of 201 observations[10] the average MAPE equals 58%, which is a rather high forecast error. However, if we look at the average MAPE for different quartiles of the data series ordering them by magnitude of the wetland value, in ascending order, we find an average MAPE of 173, 26, 16, and 19, respectively. This indicates that the fit for low wetland values is particularly poor and the fit for medium to high-valued wetlands is much more acceptable.

Subsequently, we use an $n-1$ data splitting technique to estimate 200 value transfer functions by applying the estimated parameters generated with $n-1$ observations to the omitted observation.[11] The upper panel in Figure 5 presents the observed and predicted wetland values in ascending order of observed value. This shows that for a number of observations there is a considerable difference between observed and predicted values. It also indicates that our value transfer function systematically over-predicts very low wetland values and slightly under-predicts high values.[12] The lower panel in Figure 5 presents the transfer error (defined as MAPE) associated with each observation ranked in ascending order. The overall average transfer error is 74%, which is slightly higher than the in-sample forecast error – as can be expected. The average transfer error for different quartiles of the data series ordered by magnitude of the wetland value, in ascending order, is 213, 34, 19, and 33, respectively. Slightly less than 20% of the sample has transfer errors of 10% or less, and roughly 15% of observations show transfer errors over 100%. In comparison to other value transfer exercises (reviewed in Brouwer 2000; Rosenberger and Phipps 2001) our results appear to be similar despite the relative diversity of our data in terms of the activities and services being valued, valuation techniques employed, geographic locations and socio-economic characteristics. One advantage, however, of our value transfer is

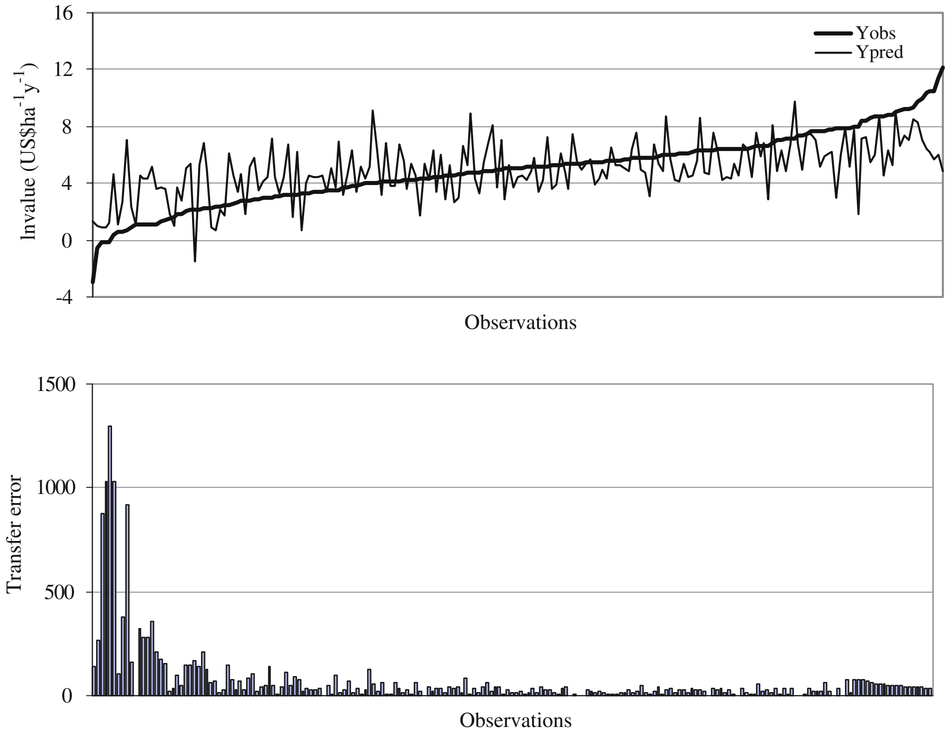

Figure 5. Observed and predicted wetland values and transfer errors, ranked in ascending order of observed wetland value.

that we have been able to include important population density and income variables that have not been included in most other studies.

An important proviso for the validity of value transfer is that sites for which a transfer is being conducted, and the method on which the valuation is based, are adequately represented in the meta-dataset (see Rosenberger and Phipps 2001). It is, however, not enough to merely count sites with specific characteristics and the number of studies using particular methods. In this case a multivariate analysis is preferable as well. We therefore regressed the exogenous variables distinguished in the meta-analysis on the transfer errors, defined as the mean absolute percentage error. This analysis shows that the transfer errors are significantly negatively correlated with the dummies for Africa, Asia, and Australasia. This result contradicts the suggestion of Rosenberger and Phipps (2001) that the accuracy of value transfer is directly related to the incidence of specific characteristics in the meta-database, because most of our observations are from North America. There is also a significant positive correlation for the replacement cost method and a similar negative correlation with net factor income methods. Finally, the transfer errors are also significantly positively correlated with the variable measuring the proportion of the wetland that is under the RAMSAR convention.

6. Conclusions

This article provides a comprehensive overview of the wetland valuation literature and has attempted to identify the important physical, socio-economic and study characteristics that determine wetland value. The wetland valuation literature has been shown to be extremely diverse in terms of values estimated, wetland types considered and valuation methods used. The value estimates produced by different valuation methodologies are not necessarily directly comparable and need to be explicitly modeled in our meta-regression. One of the key results from our meta-regression analysis is the importance of the GDP per capita and population density variables in explaining variation in wetland value. Both variables were shown to have a positive relationship with wetland value. Although such information is often not available in primary valuation studies it is suggested that future valuation meta-analyses attempt to include relevant socio-economic information from other sources in order to represent important determinants of value. Another interesting result is that CVM studies have tended, *ceteris paribus*, to produce higher value estimates than other valuation methods. This contrasts with our expectations and with the findings of Woodward and Wui (2001). In terms of the ecological and physical characteristics of wetlands, we found freshwater marshes to be valued less than other wetland types and a negative relationship between wetland size and value. Of the various wetland services that we identified, water quality improvement was found to be valued the highest. Two unexpected results from this meta-analysis were that North American wetlands and RAMSAR sites were found to be valued lower than other wetlands.

Using an $n-1$ data splitting technique we examined the robustness of using our meta-regression for out-of-sample value transfer. The resulting average transfer error is 74%, which is comparable to the transfer errors associated with other value transfer exercises in the literature. Given the high costs of performing primary valuation studies, this level of transfer error may be acceptable in considering transferred values as input in wetland conservation decisions. However, our value transfer function systematically over-predicts very low wetland values and slightly under-predicts high values. Remarkably enough, the value transfer performs better for wetlands that are located in countries not well represented in our data (Africa, Asia, and Australasia). The value transfer error is positively correlated with transfers based on the replacement cost method. The same result holds for the degree to which wetlands are RAMSAR sites. We therefore urge caution in using the results of such a meta-analysis for value transfer, particularly to policy sites for which their characteristics are not well represented in the underlying valuation studies. There is clearly still a need for more (and higher quality) primary valuation studies, particularly in developing countries.

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Notes

1. Carson et al. (1996) review 83 valuation studies for quasi-public goods from which 616 comparisons of contingent valuation (CV) and revealed preference (RP) estimates are made. The sample mean CV/RP ratio is 0.89 with a 95% confidence interval of 0.81–0.96 and a median of 0.75. While the results from this study show that RP methods produces higher value estimates than CV, it also shows that estimates from these two methods are within the same range.
2. Bateman and Jones (2003) set out expectations for the ordering of values estimated through different analytical modes of the contingent valuation and travel cost methods.
3. One particularly useful source was an annotated review of the wetland valuation literature for the period 1988–1998 by Bardecki (1998).
4. In order to compare observations in a statistical meta-analysis we required sufficient information on a number of key variables. These are: wetland value, area, type, function(s) being valued, location, year of valuation and valuation method used.
5. We used GDP deflators and purchasing power parity converters from the World Bank World Development Indicators 2002 to standardize values estimated in different years and different currencies.
6. The 1995 national per capita income level data were taken from World Bank World Development Indicators 2000, and US state level data were taken from the US Census 2000 for the US states.
7. The population densities included in our analysis represent an area of 50-km radius around each wetland site. Population and population density information was derived from CIESIN data. The process by which this data was converted to represent each wetland site in our data set is described in Wagtendonk and Omtzigt (2003).
8. A multi-level modelling (MLM) approach such as used in Brouwer et al. (1999), and Bateman and Jones (2003) was considered but not adopted. This approach incorporates natural hierarchies or levels within the data, e.g., study sites, author, method and study, allowing the (somewhat unrealistic) assumption of independence between estimates to be relaxed. MLM is, however, problematic in that it requires the use of dummy variables for each group within a level, e.g., for each author or study site. This may be feasible in reasonably limited or homogeneous data sets but less so for very diverse data (such as ours).

9. The presence of multicollinearity was tested by examining the correlation coefficients on pairs of explanatory variables and by regressing selected explanatory variables on the remaining variables.
10. For this exercise we used a slightly restricted data set as one observation for which the log value was very close to zero would otherwise have a disproportionate influence.
11. This resembles the jackknife resampling technique (see Efron 1982).
12. Again it should be noted that the primary valuation studies are highly unlikely to have produced "true" estimates of wetland value and therefore we would not necessarily expect (or want) transferred values to be exactly equal to primary valuation study results.

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